ENGINEERING DESIGN FILE

1. Project File No. WAG 5 2. Project/Task WAG 5 Operable Unit 5-12

3. Subtask WAG 5 Comprehensive Remedial Investigation/Feasibility Study

4. Title: **Sensitivity Analysis of Waste Area Group 5 Groundwater Modeling Results to Changes in Simulated Vadose Zone Sediment Thickness**

5. Summary:

This sensitivity analysis is for the comprehensive Waste Area Group (WAG) 5 remedial investigation (RI) baseline risk assessment (BRA) (Holdren et al. 1998) groundwater pathway. Two operational areas, the Auxiliary Reactor Area (ARA) and the Power Burst Facility (PBF) are included in WAG 5. A single parameter, unsaturated zone thickness, was varied to determine the sensitivity of predicted groundwater concentrations, risks, and hazard quotients to variations in the parameter. The existing WAG 5 modeling of fate and transport of contaminants in the groundwater pathway was prepared using a single average vadose zone sediment thickness value for all of WAG 5.

The existing modeling was prepared using the GWSCREEN code, which defines a parameter called *depth* to represent the unsaturated zone thickness. The unsaturated zone beneath the Idaho National Engineering and Environmental Laboratory (INEEL) is a stratified sequence of solidified basalt flows that are occasionally separated by sediment deposits of windblown, fluvial, or lacustrine origin. However, any retentive effects of basalt sequences are typically ignored in groundwater pathway risk assessments at the INEEL because significant fracturing in these brittle flows allows very rapid vertical transmission of water and water-borne contaminants in the vadose zone. Therefore, only the cumulative thickness of the interbed and surficial sediments is used to define the *depth* parameter. Beneath the INEEL, sediments typically comprise only 10% of the entire vadose zone depth.

The *depth* parameter is usually determined by summing the sediment thicknesses of the individual sedimentary interbeds beneath a given site based on subsurface lithology data gleaned from well logs and drilling notes. For WAG 5, an average cumulative unsaturated zone sediment thickness value of 5.8m was determined from available well lithology for the ARA and PBF areas. This value includes interbeds above the aquifer and the initial sediment thickness that occurs at land surface. The actual value varies spatially because the mechanisms that deposited the interbeds and those that produced basalt flows were not consistent and did not leave behind ideally uniform interbed and basalt flow thicknesses.

The impact of varying *depth* values on resulting groundwater concentration predictions was evaluated. Two additional *depth* values, an area minimum and maximum, were determined from well lithologies and then used as input in additional GWSCREEN runs. The minimum and maximum were determined to be 2.3 m and 22.5 m, respectively for WAG 5. The minimum value is approximately 60% less than the previously modeled average (5.8m) and the maximum is nearly 288% greater than the average.

The results of the sensitivity analysis show that varying the vadose zone thickness input parameter produces only very small differences in groundwater concentrations and predicted risks for those contaminants evaluated in the WAG 5 comprehensive BRA.

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Sensitivity Analysis of Waste Area Group 5 Groundwater Modeling Results to Changes in Simulated Vadose Zone Sediment Thickness

INTRODUCTION

This sensitivity analysis is for the groundwater pathway in the Waste Area Group (WAG 5) remedial investigation (RI) baseline risk assessment (BRA) (Holdren et al. 1998). Two operational areas are included in WAG 5, the Auxiliary Reactor Area (ARA) and the Power Burst Facility (PBF). A single parameter, unsaturated zone thickness, was varied to determine the sensitivity of predicted groundwater concentrations, risks, and hazard quotients to variations in the parameter. The existing WAG 5 modeling of fate and transport of contaminants in the groundwater pathway was prepared using a single average vadose zone sediment thickness value for all of WAG 5.

The existing modeling was prepared using the GWSCREEN code, which defines a parameter called *depth* to represent the unsaturated zone thickness. This parameter encompasses the total vertical distance in the unsaturated zone between the bottom of a contamination source and the top of the aquifer. The unsaturated zone beneath the Idaho National Engineering and Environmental Laboratory (INEEL) is a stratified sequence of solidified basalt flows that are occasionally separated by sediment deposits of windblown, fluvial, or lacustrine origin. However, any retentive effects of basalt sequences are typically ignored in groundwater pathway risk assessments at the INEEL because significant fracturing in these brittle flows allows very rapid vertical transmission of water and waterborne contaminants in the vadose zone. Therefore, only the cumulative thickness of the interbed and surficial sediments is used to define the *depth* parameter. Beneath the INEEL, sediments typically comprise only 10% of the entire vadose zone depth.

The *depth* parameter is usually determined by summing the sediment thicknesses of the individual sedimentary interbeds beneath a given site based on subsurface lithology data gleaned from well logs and drilling notes. For WAG 5, an average cumulative unsaturated zone sediment thickness value of 5.8m was determined from available well lithology for the ARA and PBF areas. This value includes interbeds above the aquifer and the initial sediment thickness that occurs at land surface. The actual value varies spatially because the mechanisms that deposited the interbeds and those that produced basalt flows were not consistent and did not leave behind ideally uniform interbed and basalt flow thicknesses. However, it is not unreasonable, given the paucity of subsurface lithology data, to prepare a single average value for the entire WAG.

Yet, it is also reasonable to examine how varying *depth* values will affect resulting groundwater predictions. In this analysis, two additional *depth* values, an area minimum and maximum, were determined from well lithologies then used as input in additional GWSCREEN runs. The minimum and maximum were determined to be 2.3m and 22.5 m, respectively. The minimum value is approximately 60% less than the previously modeled average (5.8m) and the maximum is nearly 288% greater than the average.

Additional GWSCREEN runs for non-decaying (i.e., nonradiological) contaminants using the minimum and maximum *depth* values indicate that predicted groundwater concentrations, risks, and hazard quotients, are not sensitive (i.e., do not change significantly) to changes in the *depth* parameter. However, the predicted time of the peak groundwater concentration changes. Literally, the unsaturated zone transit time is very sensitive to the unsaturated sediment thickness. The transit time appears to change in direct proportion to changes in the *depth* variable which correlates well with equation 21 of p.16 of the GWSCREEN User's Manual (Rood 1994). The equation shows that transit time in the unsaturated zone is directly proportional to both unsaturated zone thickness and the contaminant retardation coefficient but inversely proportional to the unsaturated pore velocity.

METHODOLOGY

To minimize the sensitivity analysis effort, only minimum and maximum *depth* values were identified to represent the unsaturated zones beneath both ARA and PBF. A group of aquifer wells in those two areas was examined for available well logs and drilling records. A variety of references were consulted for this information, including the INEEL Hydrologic Data Repository, which contains hardcopy drilling and well construction information for most wells at the INEEL; the Comprehensive Well Survey database, which contains electronic records of subsurface well lithology (DOE-ID 1994); and a database of basalt and sediment sequences prepared by the USGS (Anderson, Ackerman, and Liszewski 1996). Only sediment sequences residing above the water table were used to determine a total vadose zone thickness for each well. The resulting unsaturated zone sediment thicknesses varied considerably for a given well based on these different data sources. The variability is caused by the differences inherent in the subsurface and by the uncertainty associated with interpretations of lithologic data. For instance, some databases do not include sedimentary layers that are thinner than a specified minimum.

The minimum and maximum totals for each well included in this analysis are shown in Table 1. The minimum and maximum total for all wells included in this study are 2.3 m and 22.5 m, respectively. These are shown in Table 2 along with the base case value (average) of 5.8 m used in the RI/BRA modeling.

To further minimize the sensitivity analysis effort, only a few sites and contaminants were selected for further runs with the new *depth* values. These sites and contaminants were selected based on the potential groundwater pathway risk or hazard quotient estimated in the BRA. Sites and contaminants exceeding a 1E-07 carcinogenic risk or a 1E-03 noncarcinogenic hazard quotient were included in this study. The contaminants, site groups, and associated BRA risks and hazard quotients for groundwater ingestion are shown in Table 3 (Holdren et al. 1998). Note that arsenic was evaluated for both risk and hazard quotient, Ra-226 poses only risk, and chromium and manganese are evaluated only for hazard quotients.

To isolate the sensitivity of results to the *depth* parameter, the other input parameters used in GWSCREEN were not varied as part of this sensitivity analysis. These include the site dimensions shown in Table 4, and the contaminant-specific adsorption coefficients and initial inventories shown in Table 5. These are the same values used in the BRA base case modeling with 5.8 m unsaturated zone sediment thickness. Note that the adsorption factor in the sediment and aquifer is reduced to one-tenth the value used for the source.

Sites were grouped in the BRA based on proximity to evaluate the groundwater pathway (Holdren et al. 1998, Section 4.1). Site groups 1, 2, 3, and 6 were addressed in the sensitivity analysis. Arsenic was simulated in the groundwater model for two site groups (1 and 6); chromium for three site groups (1, 2, and 3); manganese and Ra-226 were simulated for one site group each (6 and 1, respectively). GWSCREEN version 2.4a was used to run these simulations. The code allows for direct computation of the associated risk or hazard quotient as associated with the simulated groundwater concentrations. This required separate runs for the arsenic simulations (since both risk and hazard quotient were sought) and

Table 1. Range in cumulative vadose zone sediment thicknesses in well lithology records.

Well	Minimum Vadose Zone Cumulative Sediment Thickness (m)	Maximum Vadose Zone Cumulative Sediment Thickness (m)
ARA-COR-05	2.3	3.7
ARA-MON-A-01	6.1	6.1
ARA-MON-A-02	6.1	13.1
ARA-MON-A-04	5.8	7.0
ARA-MON-A03A	13.0	15.8
ARA 2	7.6	7.6
PBF-MON-A-001	12.8	13.4
PBF-MON-A-002	8.8	10.3
PBF-MON-A-003	9.7	10.4
PBF-MON-A-004	5.5	6.1
PBF-MON-A-005	7.3	8.2
PBF CHEM. WST INJ WELL	5.2	11.3
PBF CLEAN H2O INJ WELL	4.0	6.0
SPERT-1	13.5	22.5
SPERT-2	9.7	14.3

Table-2. Cumulative vadose zone sediment thickness values used in the sensitivity analysis.

Parameter	Base Case	Case 1 (maximum)	Case 2 (minimum)
Vadose zone sediment thickness (m)	5.8	22.5	2.3
Percent change from base case	N/A	288	-60

chromium (because both trivalent and hexavalent hazard quotients were determined). Additionally, the peak groundwater concentration for manganese was found to occur after the 100-yr timeframe of interest. Therefore, an additional simulation for manganese was made to estimate the peak groundwater concentration regardless of time.

Finally, for each of the above simulations three runs were implemented: one for the base case (*depth* = 5.8 m), one for the minimum case (*depth* =2.3 m), and one for the maximum case (*depth* = 22.5 m). The base case runs ensured that the results of using an updated version of GWSCREEN (i.e., version 2.4a) do not vary significantly from the older Version 2.02 used in the BRA (Holdren et al. 1998, Section 5). The minimum and maximum runs allowed quantification of the

Table 3. Contaminants, site groups, and associated BRA risks and hazard quotients for groundwater ingestion evaluated in the sensitivity analysis.

Contaminant	Site Group	Risk	Hazard Quotient
Arsenic	Group 1	6E-06	3E-02
Arsenic	Group 6	7E-05	3E-01
Chromium(III)	Group 2	No toxicity data	2E-03
Chromium(III)	Group 3	No toxicity data	3E-03
Chromium(VI)	Group 1	No toxicity data	6E-03
Chromium(VI)	Group 2	No toxicity data	5E-01
Chromium(VI)	Group 3	No toxicity data	5E-01
Manganese	Group 6	No toxicity data	2E-03
Ra-226	Group 1	2E-07	Not applicable

Table 4. Fixed site-specific parameters applied in the sensitivity analysis.

Site Group	Source Length[a] (m)	Source Width[b] (m)	Source Thickness (m)
1	127	127	3.01
2	63.9	63.9	2.93
3	132	132	3.05
6	21.8	21.8	3.05

a. Site dimension parallel to the groundwater flow direction.
b. Site dimension perpendicular to the groundwater flow direction.

Table 5. Fixed contaminant parameters used in sensitivity analysis.

Contaminant	Site Group	Source Term Inventory (mg or Ci)	Adsorption Coefficient at Source (mL/g)	Sediment Adsorption Coefficient (mL/g)	Aquifer Adsorption Coefficient (mL/g)
Arsenic	1	6.08E+6	3.0	0.3	0.3
	6	1.76E+7			
Chromium	1	7.49E+9	1.0	0.1	0.1
	2	2.95E+8			
	3	6.88E+8			
Manganese	6	6.51E+8	50	5	5
Ra-226	1	4.46E-5	0	0	0

sensitivity of predicted concentrations, risks, and hazard quotients to changes in the unsaturated zone sediment thickness. The results of the 36 separate GWSCREEN runs are summarized in Table 6.

A cursory examination of Table 6 reveals that the magnitudes of the peak concentration, risk, and hazard quotient are not affected by changes in unsaturated zone sediment thickness for most contaminants. Only Ra-226, a decaying contaminant, is slightly affected. The results are not unexpected; all of the initial mass of the non-decaying contaminants will eventually reach the aquifer and receptor regardless of the unsaturated zone distance traveled, whereas radiological contaminants decay over time and variations in the unsaturated zone thickness affect how much contaminant remains to reach the aquifer and receptor.

For all contaminants, the time at which the simulated peak concentration arrives at the groundwater receptor is directly influenced by changes in the unsaturated zone thickness. Table 7 summarizes the peak concentration arrival times for the various sensitivity analysis cases and provides a percentage of change from the original base case arrival time. The peak concentration, maximum risk, and maximum hazard quotient occur at the same time for arsenic. This holds true also for both valence forms of chromium. Note that the percentages of change for minimum- and maximum-case arrival times in Table 7 are almost identical to the percentages of change in minimum- and maximum-case unsaturated zone thickness in Table 2.

For manganese, the peak concentration occurred after 100 years. The model was executed to estimate concentrations occurring during 100 to 130 years from the present. No manganese reaches the groundwater from the unsaturated zone during the 100 to 130-yr timeframe. In this analysis, the location of simulated receptor wells is on the downgradient edge, with respect to groundwater flow direction, of the contaminant site. Therefore, aquifer travel is minimal and the peak concentration arrival times at the receptor well (shown in Table 7) are representative of the vadose zone transit time. For manganese, the vadose zone transit times for the base, minimum, and maximum cases are 6.0E+02, 2.5E+02, 2.3E+03 yr, respectively.

The sensitivity of predicted groundwater concentrations for radiological contaminants differs from the sensitivity of nonradiological contaminants because radioactive decay is considered. For radiological contaminants, not only is the peak concentration arrival time sensitive to changes in the unsaturated zone thickness but also the peak concentration magnitude. This is reflected in the results for Ra-226 for site group 1 as shown in Table 8.

Note that the peak concentration arrival time changes in a fashion similar to the other contaminants. However, unlike the arrival time, the sensitivity of the concentration magnitude is not in direct linear proportion to the unsaturated zone sedimentary thickness but is exponentially related because it is also a function of the radioactive decay constant. Since the decay constant is specific to the contaminant, the sensitivity cannot be readily predicted nor can existing results be simply scaled to achieve new concentrations based on the amount of change in the *depth* parameter. However, the degree of sensitivity is proportional to the half-life of the contaminant. The magnitude of predicted groundwater concentrations of long-lived contaminants is less sensitive to changes in this parameter than are shorter-lived contaminants.

Table 8 also contains the carcinogenic risk value associated with each of the predicted concentrations for Ra-226. The risks are calculated using the same standard radiological carcinogenic risk formula as described in the WAG 5 BRA (Holdren et al. 1998, Section 6). Note that the 60% reduction in vadose zone thickness results in a 59% decrease in travel time but only 0.6% increase in calculated risk. Likewise, a 288% increase in sediment thickness results in a 282% increase in travel time but only a 3% decrease in groundwater ingestion risk.

Table 6. Summary of sensitivity analysis simulated peak concentrations.

Contaminant	Site Group	Base Case Sediment Thickness (5.8 m) Peak Concentration (mg/L or pCi/L)	Base Case Arrival Time (yr)	Base Case Risk	Base Case HQ	Minimum Sediment Thickness (2.3 m) Peak Concentration (mg/L or Ci/L)	Minimum Arrival Time (yr)	Minimum Risk	Minimum HQ	Maximum Sediment Thickness (22.5 m) Peak Concentration (mg/L or Ci/L)	Maximum Arrival Time (yr)	Maximum Risk	Maximum HQ
Arsenic	1	3.6E-04	6.0E+01	6E-06	3E-02	3.6E-04	2.6E+01	6E-06	3E-02	3.6E-04	2.2E+02	6E-06	3E-02
	6	4.0E-03	5.8E+01	7E-05	3E-01	4.0E-03	2.4E+01	7E-05	3E-01	4.0E-03	2.2E+02	7E-05	3E-01
Chromium(III)	2	8.5E-02	3.5E+01	NTD	2E-03	8.5E-02	1.4E+01	NTD	2E-03	8.5E-02	1.4E+02	NTD	2E-03
	3	1.0E-01	3.6E+01	NTD	3E-03	1.0E-01	1.5E+01	NTD	3E-03	1.0E-01	1.4E+02	NTD	3E-03
Chromium(VI)	1	1.2E-03	3.6E+01	NTD	6E-03	1.2E-03	1.5E+01	NTD	6E-03	1.2E-03	1.4E+02	NTD	6E-03
	2	8.5E-02	3.5E+01	NTD	5E-01	8.5E-02	1.5E+01	NTD	5E-01	8.5E-02	1.4E+02	NTD	5E-01
	3	1.0E-01	3.6E+01	NTD	5E-01	1.0E-01	1.5E+01	NTD	5E-01	1.0E-01	1.4E+02	NTD	5E-01
Manganese	6	9.6E-03	5.9E+02	NTD	2E-03	9.6E-03	2.5E+02	NTD	2E-03	9.6E-03	2.3E+03	NTD	2E-03
Ra-226	1	3.1E-14	2.4E+01	2E-07	2E-03	3.1E-14	9.9E+00	2E-07	NTD	3.0E-14	9.3E+01	2E-07	2E-03

HQ = hazard quotient

NTD = no toxicity data

NA = not applicable.

Table 7. Summary of percentage changes in peak concentration arrival times.

Contaminant	Site Group	Base Case (*depth* = 5.8m) Arrival Time (yr)	Minimum Case (*depth* = 2.3 m) Arrival Time (yr)	Percent Change of Minimum from Base Case	Maximum Case (*depth* = 22.5 m) Arrival Time (yr)	Percent Change of Maximum from Base Case
Arsenic	1	6.03E+01	2.60E+01	-5.7E+01	2.24E+02	2.7E+02
Arsenic	6	5.84E+01	2.41E+01	-5.9E+01	2.22E+02	2.8E+02
Chromium(III)	2	3.59E+01	1.49E+01	-5.9E+01	1.36E+02	2.8E+02
Chromium(III)	3	3.63E+01	1.53E+01	-5.8E+01	1.36E+02	2.8E+02
Chromium(VI)	1	3.63E+01	1.53E+01	-5.8E+01	1.36E+02	2.8E+02
Chromium(VI)	2	3.59E+01	1.49E+01	-5.9E+01	1.36E+02	2.8E+02
Chromium(VI)	3	3.63E+01	1.53E+01	-5.8E+01	1.36E+02	2.8E+02
Manganese	6	5.97E+02	2.50E+02	-5.8E+01	2.25E+03	2.8E+02
Ra-226	1	2.43E+01	9.90E+00	-5.9E+01	9.27E+01	2.8E+02

Table 8. Summary of percentage changes in peak concentration and associated risk for Ra-226 in site group 1.

Case	Peak Concentration (pCi/L)	Change in Peak Concentration from Base Case (%)	Arrival time (yr)	Change in Arrival Time from Base Case (%)	Risk	Change in Risk from Base Case (%)
Base	3.11E-2	NA	24.3	NA	1.959E-07	NA
Minimum	3.13E-2	0.6	9.9	-59	1.972E-07	6.4E-01
Maximum	3.02E-2	-2.9	92.7	282	1.903E-07	-2.9E+00

As a final exercise in this sensitivity analysis, the output from two versions of GWSCREEN were compared. Version 2.4a is the most current version of this model. Risk and hazard quotient values in the original modeling effort for the WAG 5 BRA were prepared outside of the GWSCREEN code. Therefore, the risks and hazard quotients from the GWSCREEN Version 2.4a output files are not compared against the GWSCREEN output files (Holdren et al. 1998, Appendix D) in the BRA. However, peak concentrations and corresponding arrival times for the two versions of the base case GWSCREEN results are presented in Table 9.

The same parameter values were used for both versions of the code. Note that for almost every site group and contaminant the resulting peak concentrations and arrival times are nearly identical. Small differences are expected and acceptable because Version 2.4a is a refinement of the previous version. The results in Table 9 are identical to the first significant digit.

Table 9. Comparison of GWSCREEN Version 2.4a output to GWSCREEN Version 2.02 output.

		Maximum Concentration for Base Case (*depth* = 5.8 m)			Arrival Time (yr)		
Contaminant	Site Group	Version 2.02	Version 2.4a	Difference (%)	Version 2.02	Version 2.4a	Difference (%)
Arsenic	1	3.62E-04	3.62E-04	0.0	6.03E+01	6.03E+01	0.0
	6	3.80E-03	3.96E-03	4.2	5.76E+01	5.84E+01	1.4
Chromium	1	1.15E-03	1.15E-03	0.0	3.63E+01	3.63E+01	0.0
	2	8.46E-02	8.47E-02	0.1	3.58E+01	3.59E+01	0.3
	3	1.00E-01	1.00E-01	0.0	3.63E+01	3.63E+01	0.0
Manganese	6	9.15E-03	9.55E-03	4.4	5.87E+02	5.97E+02	1.7
Ra-226	1	3.14E-02	3.11E-02	-1.0	2.43E+01	2.43E+01	0.0

SUMMARY

In conclusion, this sensitivity analysis of the unsaturated zone sediment thickness in the WAG 5 groundwater modeling examined a minimum sediment thickness value (2.3 m) and a maximum value (22.5 m) and compared results with those made with the average unsaturated zone sediment thickness (5.8 m).

For non-decaying contaminants, the results indicate that the magnitude of the peak groundwater concentrations are not affected by changes in this parameter. The time at which the peak concentration arrives at the receptor location is, however, strongly dependent on this parameter and appears to be directly proportional. Arrival times with the minimum thickness value were all about 60% less than the base case while those with the maximum thickness were all about 280% greater. These percentages correlate well with the differences in minimum and maximum unsaturated zone sediment thickness from the average value. However, the magnitudes of the peak concentrations and, hence, the associated risks and hazard quotients, of non-decaying contaminants, remain the same regardless of changes to the *depth* parameter. It may be possible to scale the arrival times of existing results of non-decaying contaminants by using a simple scaling factor based on changes in the unsaturated zone thickness.

For radiological (i.e., decaying) contaminants, the magnitude of the predicted peak groundwater concentration is affected by the value of the unsaturated zone thickness, but not linearly. Decay of the contaminant is a function of time, and the amount of time the contaminant spends in the unsaturated zone is proportional to the depth of the unsaturated zone. Arrival times of peak concentrations of radiological contaminants appear to be affected in a manner similar to non-radiological contaminants. For Ra-226, the radiological contaminant included in this analysis, a 60% decrease in the vadose zone thickness resulted in less than 1% increase in the associated residential groundwater ingestion risk.

Finally, the differences between GWSCREEN Version 2.02 and Version 2.4a discovered in this analysis appear to be insignificant.

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ENGINEERING DESIGN FILE

Project/Task: Comprehensive Risk Assessment

Subtask: WAG-05

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TITLE: Groundwater Risk Assessments for the PBF Warm-Waste Injection Well (PBF-05), the PBF Corrosive-Waste Injection Well (PBF-15), and the SPERT-IV Leach Pond (PBF-22)

SUMMARY

Health risk evaluations for three sites within the Waste Area Group 5 Operable Units (OUs) 5-08 and 5-09 were reassessed for the 100-year residential groundwater pathway. The three sites include the PBF Warm-Waste Injection Well (PBF-05), the PBF Corrosive-Waste Injection Well (PBF-15), and the SPERT-IV Leach Pond (PBF-22). Previous investigations revealed no unacceptable risks at these sites; however, agencies reviewing the previous efforts have recommended further analysis. The sites were re-examined for groundwater ingestion risk using the screening code GWSCREEN with new input values based on more complete sampling results and more reasonable conceptual models.

The injection wells, PBF-05 and PBF-15, were each modeled using the GWSCREEN pond release option. This allowed simulation of contaminant transport via the injected waste water that occurred during the wells' operational periods. Leaching of the contaminants due to infiltration following the end of discharge operations was also incorporated into these simulations. The SPERT-IV leach pond, PBF-22, was modeled as a buried source. Release of contaminants occurred as a result of leaching due to infiltration. Due to a lack of discharge data, no attempt was made to incorporate discharged waste water as a means of PBF-22 contaminant transport.

Although Sr-90 (PBF-05), hydrazine (PBF-15), U-234 (PBF-22), and U-238 (PBF-22) failed an initial risk screening of peak contaminant concentration, further investigation revealed that none of the contaminants at any of the three sites exceed the screening criteria associated with the 100-year residential scenario; therefore, no unacceptable 100-year residential health risks were identified for the sites PBF-05, PBF-15, and PBF-22.

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Groundwater Risk Assessments for the PBF Warm-Waste Injection Well (PBF-05), the PBF Corrosive-Waste Injection Well (PBF-15), and the SPERT-IV Leach Pond (PBF-22)

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June 24, 1996

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ACRONYMS

COC	contaminant of concern
INEL	Idaho National Engineering Laboratory
INWMIS	INEL Non-Radiological Waste Management Information System
MCL	maximum contaminant level
OU	Operable Unit
PBF	Power Burst Facility
RWMC	Radioactive Waste Management Complex
RWMIS	Radioactive Waste Management Information System
SPERT-IV	Special Power Excursion Reactor Test IV

Groundwater Risk Assessments for the PBF Warm-Waste Injection Well (PBF-05), the PBF Corrosive-Waste Injection Well (PBF-15), and the SPERT-IV Leach Pond (PBF-22)

INTRODUCTION

Groundwater ingestion risks for three sites, the Power Burst Facility Operable Unit 5-08 (PBF-05/OU 5-08) Warm Waste Injection Well, PBF-15/OU 5-08 Corrosive Waste Injection Well, and PBF-22/OU 5-09 SPERT-IV Leach Pond, were re-evaluated based on agency comments regarding initial groundwater risk assessments documented in the OU 5-08/5-09 Track 2 Summary Report. Agency comments raised concerns about the uncertainty of the quantities of waste disposed at PBF-15, the PBF-15 source zone definition, the use of incomplete sampling results in the PBF-22 analysis, and the use of unvalidated sample data in the PBF-05 initial screening.

The model GWSCREEN was used to re-examine the three sites using updated input parameters. PBF-05 and PBF-15 were each modeled with GWSCREEN using the pond release model based on known discharged quantities of liquid waste. This allowed simulation of advective contaminant transport from the source zone to the saturated zone, therefore accounting for the waste water injected during the operational period of the well. It also included leaching of contaminants due to infiltration from natural precipitation following the end of the well operational period. Although PBF-22 was an actual waste water pond, no information could be found on liquid discharge rates to the pond. As a result, PBF-22 was modeled as a buried source with leaching due only to natural precipitation. Due to a lack of discharge data, discharged waste water as a means of contaminant transport was not incorporated, and no attempt was made to model PBF-22 with the pond release model option.

In each case, source-term definitions and other GWSCREEN options differed from the earlier Track 2 evaluation. The new screening effort showed that at PBF-05, -15, and -22, no unacceptable health risks for the 100-year residential scenario were identified for any contaminants at these sites.

The Track 2 Summary Report (Hillman-Mason et al., 1994) contains calculated risks for these three sites. These risks were calculated from the peak groundwater concentration, regardless of when this peak occurred (Hillman-Mason et al., 1994) (Table 1). The set of COCs that were evaluated in both this analysis and the Track 2 investigations for PBF-05 include Co-60, Cs-137, H-3, and Sr-90; for PBF-15 they include Cr-III, hydrazine, and zinc; for PBF-22, they were Am-241, Arochlor-1248, Arochlor-1254, chromium-III, Co-60, Cs-137, mercury, Pu-239, silver, Sr-90, U-234, and U-238.

For the PBF-05 and PBF-15 COCs, peak groundwater concentrations and associated health risks were higher in the results of this analysis and, in some cases, higher by several orders of magnitude relative to the earlier Track 2 investigation; however, COC peak groundwater concentrations and

Table 1. Track 2 Summary Report risk results (from Hillman-Mason et al., 1994).

Site	Contaminant	Source Term	Calculated Peak Risk or Hazard	Scenario Risk or Hazard Calculated for
PBF-05	Co-60	0.3 pCi/g	Risk=9E-126	Residential
	Cs-137	0.5 pCi/g	Risk=6E-61	Residential
	H-3	80.0 pCi/g	Risk=4E-226	Residential
	Sr-90	0.004 pCi/g	Risk=1E-10	Residential
PBF-15	Cr-III	17.96 mg/kg	Hazard quotient=0.0003	Residential
	Hydrazine	0.18 mg/kg	Risk=8E-11	Residential
	Zn	5.35 mg/kg	Hazard quotient=3.8E-5	Residential
PBF-22	Arochlor-1254	0.025 mg/kg	Risk=4E-5	Occupational
	Cr-III	18.8 mg/kg	Hazard quotient=0.0006	Residential
	Cs-137	0.87 pCi/g	Risk=2E-5	Residential
	Hg	0.06 mg/kg	Hazard quotient=0.0007	Residential

associated health risks were all found to be much lower for PBF-22, relative to the Track 2 results.

Contaminant travel times for the Track 2 investigation differed greatly from those found in this analysis. For the two injection wells, the Track 2 had treated them as buried waste sources; this analysis treats them as disposal ponds. The additional advective transport associated with the disposal pond concept caused contaminants to travel to the receptor at much higher rates. For the two injection wells, this analysis produced vadose zone transit times for each COC that ranged from one to two orders of magnitude less than the Track 2 travel times. For the leach pond, this analysis produced vadose zone travel times that ranged from 0.5 to 4 times greater than the Track 2 investigation.

The Track 2 investigation source terms for the two injection wells were determined as concentrations based on estimated source volumes and known discharge data. In this analysis, a different modeling approach allowed the injection well source terms to be defined by the known data on disposed contaminants. For both the Track-2 and this analysis, soil concentrations from several sampling campaigns were used to define source terms for PBF-22.

To provide a better understanding of the COC travel times, peak occurrences, and rates of COC dissipation, all COCs for all sites in this analysis were initially analyzed for risk or hazard quotient based on the peak groundwater concentration. To qualify for the 100-year residential risk analysis, the COCs had to first fail an initial screening process in which either risks or hazard quotients, calculated from peak groundwater concentrations, were compared to screening criteria of 1E-07 for carcinogens and 0.1 hazard quotient for noncarcinogens. These screening criteria of 1E-07 for carcinogens and 0.1 hazard quotient for noncarcinogens are used for screening individual COCs when preparing cumulative risk assessments (Burns, 1995). Four contaminants (Sr-90 at PBF-05, hydrazine at PBF-15, and U-234 and U-238 at PBF-22) failed this initial screening and were analyzed in detail for 100-year residential health risks. None of these were shown to pose any unacceptable risks to future residents. This report describes the methodology and results from both the initial screening process and the 100-year residential risk analysis.

SITE DESCRIPTIONS

PBF-05

The PBF Warm Waste Injection Well is located 25 m (82 ft) south of the PBF reactor building (PBF-620) in the PBF Reactor Area. The 25.4-cm (10-in.) diameter well was drilled and completed to a depth of 33.5 m (110 ft) below land surface, approximately 105 m (345 ft) above the regional aquifer. The well operated from 1973 to 1984, before being capped and abandoned. From 1973 to 1980, the well received low-level radioactive waste water from the PBF-620 warm waste sump, which collected low-activity fluids from various floor and equipment drains. The well also received uncontaminated raw cooling water from the plant equipment's utility cooling system. The reported average annual discharge rates for these two streams were 1.2×10^6 L/yr (3.17×10^5 gal/yr) and 6.1×10^6 L/yr (1.61×10^6 gal/yr), respectively (Hillman-Mason et al., 1994). From 1981 to 1984, only uncontaminated raw cooling water was discharged to the well. In 1984, the well was sealed and capped.

PBF-15

The PBF corrosive-waste injection well is located 55 m (180 ft) northeast of the warm waste injection well in the PBF Reactor Area. This 10.2-cm (4-in.) diameter well was drilled to 35 m (115 ft) below land surface, approximately 103.6 m (340 ft) above the aquifer. The well received an average of 1.1×10^6 L/yr (2.91×10^5 gal/yr) of chemical wastewater from 1971 to 1978 from the draining of cooling systems and the regeneration of demineralizers. Discharge to the well was discontinued in 1979; the well was then plugged and abandoned.

PBF-22

The SPERT-IV leach pond is located 83 m (272 ft) south of the SPERT-IV reactor building (PBF-613) and was used for the disposal of contaminated waste effluent from the reactor building and waste holdup tank and also received chemical waste byproducts of water softening and demineralization activities; however, there is no record of waste stream constituents and quantities discharged. The pond is believed to have received reactor operations-related waste water from 1961, the startup of the SPERT-IV reactor, until 1970. From 1970 to 1978, the facility was used for a variety of material tests, and some discharges to the pond may have occurred although the reactor had been removed (Suckel, 1986). However, no documentation exists of any quantities disposed at the pond during this period. Decommissioning was completed in 1979. The SPERT-IV facility was used for non-reactor functions from 1979 to 1981, producing an unrecorded quantity of pond discharge. In 1983, the Idaho Chemical Processing Plant was temporarily unable to process and dispose of reactor liquid waste. During this period, the PBF reactor was operating and the SPERT-IV leach pond was used to dispose of resulting liquid wastes. The drained reactor coolant system liquids were first recycled and filtered through ion-exchange columns and placed in tanker trucks. About 62,000 L (16,400 gal) of the treated wastes were emptied from the trucks into the leach pond (Suckel, 1986). According to the 1986 Installation Assessment Report (EG&G Idaho, 1986), the leach pond soil, contaminated by the PBF reactor liquid waste, was removed and sent to the Radioactive Waste Management Complex (RWMC). The piping systems conveying effluent to PBF-22 have since been removed.

INITIAL SCREENING OF PBF-05 AND PBF-15 INJECTION WELLS

Conceptual Model

The analyses performed for this study of PBF-05 and PBF-15 were based on a conceptual model that combines the features of a buried source and a standing water pond. Although PBF-05 and PBF-15 were injection wells, the GWSCREEN pond release model option allows simulation of the advective transport of contaminants from the unsaturated zone to the saturated zone accounting for the added liquid effluent discharged to the disposal wells during the wells' operational periods.

The pond option also allows simulation of the leaching of contaminants via infiltration of rainfall and snowmelt. This involves the relatively slow removal of contaminants that have remained in the unsaturated subsurface following the wells' operational periods. In this case, infiltrating water moves under unit hydraulic gradient conditions through the source. A graphical representation of the injection well conceptual model is shown in Figure 1.

Source Zone Definitions

For the two injection wells, source zones were defined by the area that would be created when the injected flow passed through sedimentary interbeds. This was determined mathematically as the average annual injection flow rate divided by the representative saturated hydraulic conductivity of the sedimentary interbed material. Dimensionally,

$$\text{Flow Rate/Saturated Hydraulic Conductivity} = (m^3/yr)/(m/yr) = m^2$$

corresponds to units of area. This is an acceptable method for determining extent of lateral subsurface spreading (personal communication, A. S. Rood, 1996). The source zone width and length were then both set equal to the square root of this area. A source zone vertical thickness of 1.0 m was used in the analysis of both wells. Due to the lack of any data on vertical characterization of the subsurface source, this value was chosen as a simplifying, best-guess estimate. For the two injection wells, contaminant concentrations at the start of the simulations were assumed to be zero. Known data on total quantities of COC disposed were used as input to the wells during the simulations.

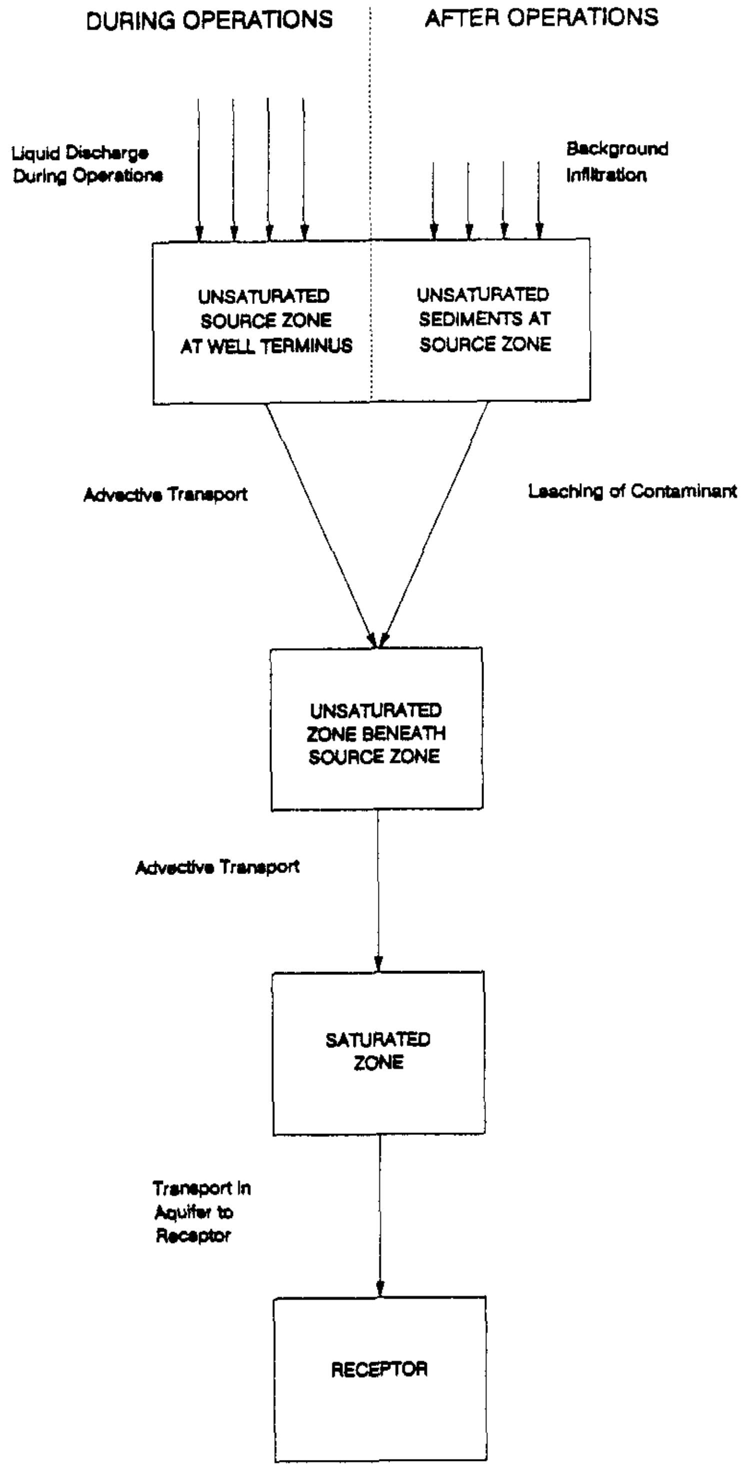

Figure 1. Conceptual model for injection wells PBF-05 and PBF-15.

Liquid Discharge Rates

Volumes of effluent injected into PBF-05 are documented in the Radioactive Waste Management Information System (RWMIS) and the INEL Non-Radiological Waste Management Information System (INWMIS) databases maintained at the INEL. Only the INWMIS database contained release data for PBF-15 since this was a non-radiological waste injection well. Records of effluent discharges were used to calculate the annual average flow rate. This value was then used as the facility water flux to the source, a required input parameter for the pond release model of GWSCREEN. Liquid discharge annual volumes and averages for the PBF-05 and PBF-15 injection wells are given in Table 2.

Contaminants of Concern

Four contaminants of concern were incorporated in the PBF-05 initial screening analysis: Co-60, Cs-137, H-3, and Sr-90. These COCs were selected based on information contained in the Track 2 Summary Report (Hillman-Mason et al., 1994) and from the RWMIS database (DOE-ID, 1996b). COCs for the PBF-15 risk assessment include chromium-III, hydrazine, and zinc.

Model simulation times begin the first year of well operation (1973 for PBF-05; 1971 for PBF-15). Initial contaminant inventories in the injection well source zones were set to zero. The GWSCREEN input parameter RMI [rate of mass input (Ci/yr or mg/yr)], was calculated by dividing the total known quantity of discharged COC by the known total operational time for which the COC was discharged at the facility. The total operational times (GWSCREEN parameter TOPER) used in these simulations are specific to each COC and were determined from RWMIS and INWMIS databases. Due to a lack of data, the values assigned to TOPER for hydrazine (PBF-15) and zinc (PBF-15) were 1 year and 3 years, respectively. Although wastewater discharge data exist for the 8 years (1971 to 1978) as shown in Table 2, concentrating all of the release of these two COCs into 1 year for hydrazine and 3 years for zinc conservatively increases the chance for detection at the receptor location. A summary of COC discharged quantities, source volume input rates, and operating years for these two sites is given in Table 3.

INITIAL SCREENING OF PBF-22 LEACH POND

Conceptual Model

The PBF-22 leach pond was modeled as a buried waste source that leaches contaminants via natural percolation following the end of the operational period for this facility. This site was not modeled with the GWSCREEN pond release option because historical liquid discharge data were insufficient. Only one record of 62,000 L (16,400 gal) of reactor coolant discharge could be found. The pond's conceptual model is shown in Figure 2.

Source Zone Geometry

The source zone for PBF-22 was determined from examination of an older sampling network grid (Hardy and Stanisich, 1989) from which an equivalent rectangular source area was determined. The horizontal area was assumed to be the entire surface area of the pond. The source zone

Table 2. Liquid discharge rates to injection wells PBF-05 and PBF-15.

PBF-05 Warm-Waste Injection Well						PBF-15 Corrosive-Waste Injection Well		
INWMIS Database			RWMIS Database			INWMIS Database		
Year	Volume (m^3)	Average (m^3/yr)	Year	Volume (m^3)	Average (m^3/yr)	Year	Volume (m^3)	Average (m^3/yr)
1973	3.187E+03	3,187	1973	8.147E+02	815	1971	6.550E+02	655
1974	2.055E+03	2,621	1974	1.657E+03	1,236	1972	1.113E+03	884
1975	5.290E+03	3,511	1975	1.578E+03	1,350	1973	6.018E+02	790
1976	1.430E+04	6,208	1976	1.710E+03	1,440	1974	1.188E+03	889
1977	8.379E+03	6,642	1977	2.143E+03	1,581	1975	1.089E+03	929
1978	6.860E+03	6,679	1978	6.567E+02	1,427	1976	9.765E+02	937
1979	7.783E+03	6,836	1979	4.598E+02	1,288	1977	1.450E+03	1,010
			1980	5.163E+01	1,134	1978	1.321E+03	1,049
Total	4.785E+04	6,836	Total	9.071E+03	1,134	Total	8.394E+03	1,049

Table 3. Contaminant discharge rates to injection wells PBF-05 and PBF-15.

Contaminant of concern	Type of COC	Total activity or mass discharged (Ci or mg)	Years Disposed	Total Years	Rate of input (Ci/yr or mg/yr)	Reference
PBF-05						
Cs-137	radionuclide	3.020E-O1	1975-80	6	0.05033	RWMIS (DOE-ID, 1996b)
Sr-90	"	1.804E-03	1975-80	6	0.0030	RWMIS (DOE-ID, 1996b)
Co-60	"	2.749E-03	1975-78	4	0.00069	RWMIS (DOE-ID, 1996b)
H-3	"	2.10E-02	1974-80	7	0.00300	RWMIS (DOE-ID, 1996b)
PBF-15						
Cr-III	non-carcinogen	3.014E+05	1971-78	8	37,678	INWMIS (DOE-ID, 1996a)
Hydrazine	carcinogen	1.000E+03	1971-78?	1	1,000	INWMIS (DOE-ID, 1996a)
Zn	non-carcinogen	1.900E+04	1971-78?	3	6,333	INWMIS (DOE-ID, 1996a)

thickness would normally be based on the vertical extent of contamination as determined from sampling efforts. However, this site has undergone several soil and subsurface sampling campaigns, and none of the efforts have been able to satisfactorily determine either the vertical or lateral extent of contamination. Resulting sample concentrations were used to define contaminant source terms; these are discussed in the following Contaminants of Concern section. The 3-m value used as the source zone thickness for this site corresponds roughly to the average depth of surficial sediments at this location (Hardy and Stanisich).

Liquid Discharge Rates

Insufficient records of volumes or rates of liquid discharge to PBF-22 precluded determination of an annual average facility flux to the contaminant source and, hence, only the GWSCREEN buried source model was used in this analysis.

Contaminants of Concern

For the initial screening analysis of PBF-22, it was assumed that all COCs were in place in the source zone at the end of the last known contaminant discharge period (1983), which corresponds to the start of the simulated leaching process. COCs were selected based on records from the several sampling events at this site. The detected contaminants were not consistent between sampling events; therefore, all detected contaminants were assumed uniformly distributed throughout the source volume at the maximum detected concentration. The PBF-22 COCs include silver, Arochlor-1248 and Arochlor-1254 (polychlorinated biphenyls), Am-241, Co-60, chromium-III, Cs-137, mercury, Pu-239, Sr-90, U-234, and U-238. Table 4 presents the sampled concentrations and calculated initial inventories for PBF-22.

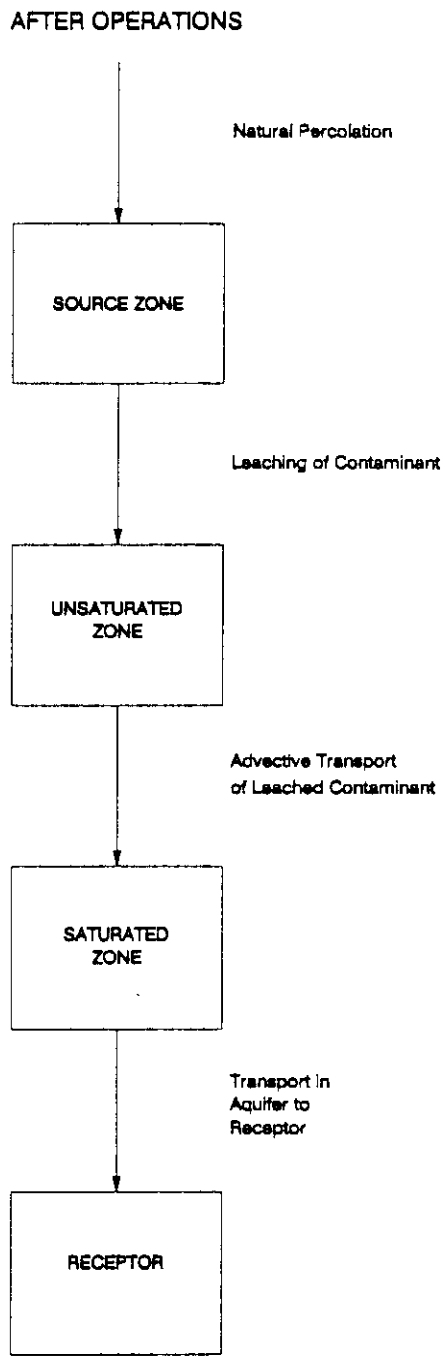

Figure 2. Conceptual model for leach pond PBF-22.

Table 4. Contaminants of concern for site PBF-22.

Contaminants of Concern	Type of COC	Soil Concentration	Estimated Initial Mass or Activity	Sampling Date (Reference: Hillman-Mason et al., 1994)
Ag	noncarcinogen	0.87 mg/kg	1.659E+07 mg	1994
Arochlor-1248	"	0.025 mg/kg	4.577E+05 mg	1994
Arochlor-1254	"	0.78 mg/kg	1.488E+07 mg	1988
Cr-III	"	147 mg/kg	2.803E+09 mg	1988
Hg	"	0.11 mg/kg	2.098E+06 mg	1988
Am-241	radionuclide	0.03 pCi/g	5.721E-04 Ci	1994
Co-60	"	2.25 pCi/g	4.291E-02 Ci	1985
Cs-137	"	11.1 pCi/g	2.117E-01 Ci	1985
Pu-239	"	0.02 pCi/g	3.814E-04 Ci	1994
Sr-90	"	5.4 pCi/g	1.029E-01 Ci	1985
U-234	"	2.85 pCi/g	5.435E-02 Ci	1994
U-238	"	1.2 pCi/g	2.289E-02 Ci	1994

COMMON PARAMETERS FOR PBF-05, -15, AND -22

Other Input Parameters

Table 5 contains transport-specific data for each COC including radiological half-lives, molecular weights, and sorption coefficients. The COC-specific sorption coefficients used in this analysis were taken from default values provided in Track 2 assessment guidance (DOE-ID, 1994b). Solubility limits were assigned a high value (1E+06 mg/L) to minimize the effects of solubility limits on contaminant transport. Other contaminant-specific data such as toxicity data in the form of carcinogenic risk slope factors and noncarcinogenic reference doses are presented in Table 6. Carcinogenic slope factors were taken from the most recent supplement to the HEAST tables (EPA, 1995). There are no established EPA toxicity data available for Arochlor-1248; therefore it was not quantitatively evaluated in this risk assessment. Also shown, where available, are the maximum contaminant levels (MCLs) for each COC, which are taken from the EPA Clean Water Act drinking water standards.

Unsaturated Zone Thickness

For the GWSCREEN simulations, the unsaturated zone was assumed to consist only of the sedimentary interbeds between the contaminant release point and the top of the aquifer. INEL Track 2 guidance (DOE-ID, 1994b) recommends ignoring the travel time for water to move vertically through basalt layers separating the sedimentary interbeds. Lithologic logs of wells in the vicinity of the sites were used to estimate total interbed thicknesses.

Table 5. Default values for contaminant transport properties.

Site	Contaminant	Molecular Weight (g/mol)	Radiological Half-Life (yr)	Solubility Limit (mg/L)	Sorption Coefficients Source (mL/g)	Sorption Coefficients Unsaturated (mL/g)	Sorption Coefficients Saturated (mL/g)
PBF-05	Co-60	58.93	5.27	1.0E+06	60	60	10
	Cs-137	132.9	30.3	1.0E+06	280	280	500
	H-3	3	12.3	1.0E+06	0.00	0.00	0.00
	Sr-90	87.62	29.1	1.0E+06	15	15	24
PBF-15	Cr-III	51.99	NA	1.0E+06	70	70	1.2
	Hydrazine	32	NA	1.0E+06	0.0003	0.0003	0.0003
	Zn	65.39	NA	1.0E+06	200	200	16
PBF-22	Ag	107.8	NA	1.0E+06	90	90	90
	Am-241	241	432	1.0E+06	1900	1900	340
	Arochlor-1248	372	NA	1.0E+06	100	100	10
	Arochlor-1254	372	NA	1.0E+06	100	100	10
	Co-60	58.93	5.27	1.0E+06	60	60	10
	Cr-III	51.99	NA	1.0E+06	1.2	1.2	1.2
	Cs-137	132.9	30.3	1.0E+06	280	280	500
	Hg	200.9	NA	1.0E+06	100	100	100
	Pu-239	239	2.41E+4	1.0E+06	550	550	22
	Sr-90	87.62	29.1	1.0E+06	15	15	24
	U-234	234	2.45E+5	1.0E+06	35	35	6
	U-238	238	4.47E+9	1.0E+06	35	35	6

a. NA = not applicable.
b. No molecular weight or sorption coefficient data are available for Arochlor-1248. Arochlor-1254 molecular weight and sorption coefficients were assumed for Arochlor-1248.

Table 6. Default values for contaminant toxicity properties.

Site	Contaminant	Type of COC	Maximum Contaminant Level	Slope Factor (risk/pCi) or $(mg/kg/d)^{-1}$	Reference Dose [RfD] (mg/kg/d)	Slope Factor and RfD References
PBF-05	Co-60	radionuclide	218 pCi/L	1.89E-11	NA	EPA, 1995
	Cs-137	"	119 pCi/L	3.16E-11	NA	EPA, 1995
	H-3	"	20,000 pCi/L	7.15E-14	NA	EPA, 1995
	Sr-90	"	8 pCi/L	5.59E-11	NA	EPA, 1995
PBF-15	Cr-III	noncarcinogen	100 μg/L	NA	1.0	EPA, 1990
	Hydrazine	carcinogen	400 μg/L	3.0	NA	IRIS, 1996
	Zn	noncarcinogen	ND	NA	0.3	EPA, 1995
PBF-22	Ag	"	ND	NA	5E-3	EPA, 1995
	Arochlor-1248	"	ND	NA	ND	IRIS, 1996
	Arochlor-1254	"	ND	NA	2E-5	IRIS, 1996
	Cr-III	"	100 μg/L	NA	1.0	EPA, 1990
	Hg	"	2 μg/L	NA	3E-4	EPA, 1994
	Am-241	radionuclide	6.45 pCi/L	3.28E-10	NA	EPA, 1995
	Co-60	"	218 pCi/L	1.89E-11	NA	EPA, 1995
	Cs-137	"	119 pCi/L	3.16E-11	NA	EPA, 1995
	Pu-239	"	64.9 pCi/L	3.16E-10	NA	EPA, 1995
	Sr-90	"	8 pCi/L	5.59E-11	NA	EPA, 1995
	U-234	"	13.9 pCi/L	4.44E-11	NA	EPA, 1995
	U-238	"	14.6 pCi/L	4.27E-11	NA	EPA, 1995

NA = Not applicable; ND = No toxicity or carcinogenic effects data available or No established drinking water standards.

The waste injection zones at wells PBF-05 and PBF-15 were located 33.5 m (110 ft) and 35.4 m (116 ft) below land surface, respectively. The nearest well to the two PBF injection wells with a reliable lithologic log down to the aquifer is PBF-MON-A-001; the well log indicates a sum total interbed thickness of 10.4 m (34 ft) from 35.4 m (116 ft) below land surface, the approximated depth of the injection zone, down to the top of the aquifer. This value was used for both injection well simulations. Interbed thicknesses of other wells in the surrounding area (PBF-MON-A-002, PBF-MON-A-005, SPERT-I, and USGS 20) range from 7.3 m to 11.9 m (24 ft to 39 ft) (DOE-ID, 1994a). When considered with well PBF-MON-A-001, the average interbed thickness of these wells is 10.2 m (33.5 ft), which is very close to the 10.4 m (34 ft) value used.

At the SPERT-IV leach pond (PBF-22), the total interbed thickness used for the simulations was also 10.4 m (34 ft). This is the total interbed thickness between land surface and the top of the aquifer (not including surficial sediments) according to the lithologic log for well PBF-MON-A-002, which is approximately 805 m (2,640 ft) from PBF-22. Interbed thicknesses of other wells in the surrounding area (PBF-MON-A-003, PBF-MON-A-004, and SPERT-I) range from 6.1 m to 14.4 m (20 ft to 47 ft) (DOE-ID, 1994a). When considered with well PBF-MON-A-002, the average interbed thickness of these wells is 10.5 m (34.5 ft), which is very close to the 10.4-m (34-ft) value used. It is purely coincidence that the same value was used for the injection wells and the leach pond, because the interbed thickness at PBF-22 was based on different wells and accounted for sediments above the injection zone depth of the wells.

Aquifer formation hydraulic properties used in this analysis were either derived from Track 2 defaults (DOE-ID, 1994b) or were obtained from INEL-specific data presented in the GWSCREEN documentation (Rood, 1994). These include aquifer and interbed porosities and bulk densities, aquifer thickness, saturated pore velocity, and longitudinal and transverse dispersivities. Values used in the GWSCREEN simulations are presented in Table 7. Site-specific media parameters are described below.

Table 7. Track 2 default parameters for media and other properties.

Property	GWSCREEN Variable	Default Value	Units
Bulk density source zone	RHOS	1.5	g/cm^3
Bulk density unsaturated zone	RHOU	1.5	g/cm^3
Bulk density aquifer	RHOA	1.9	g/cm^3
Porosity of aquifer	PHI	0.1	m^3/m^3
Longitudinal dispersivity	AX	9	m
Transverse dispersivity	AY	4	m
Vertical dispersivity	AZ	0.4	m
Saturated pore velocity	VX	570	m/yr
Well screen thickness	THICK	15	m
Evaporation loss rate constant	EVAP	0	1/yr
Other loss rate constant	RC2	0	1/yr
Net percolation rate	PERC	0.1	m/yr

The volumetric moisture content used in the GWSCREEN simulations of the injection wells was obtained from the INEL sediment moisture characteristic curve by assuming that unit hydraulic gradient conditions existed. Therefore, during operations, the volumetric moisture content of the source zone and the unsaturated zone was set equal to the average porosity of the INEL sediment samples (48.7%) described in the GWSCREEN manual (Rood, 1994). This is because the source zones were sized assuming that saturated conditions existed beneath the injection zones for PBF-05 and PBF-15. The moisture content of the source zone after operations reverts to a value of 0.343, which corresponds to the background infiltration rate of 0.1 m/yr under unit gradient conditions. The moisture content of the unsaturated zone does not change in the GWSCREEN model. This means that the travel time through the unsaturated zone, after drainage of the injected waste water, will be longer by the ratio of the saturated moisture content to the background moisture content (1.4 in this case).

Other Risk Analysis Parameters

In addition to the known COCs, the risk analysis of certain radionuclide COCs requires consideration of radiological progeny. Specifically, PBF-22 sampling results indicate that the pond contains the alpha-emitting COCs Am-241, Pu-239, U-234, and U-238, which all decay to unstable daughter products that can contribute significantly to the overall risk posed by the parent nuclide. The decay chain for Am-241 includes progeny considered to contribute significantly to total risk, which were therefore included in this analysis; these are Np-237, U-233, and Th-229. The Pu-239 analysis included daughters U-235, Pa-231, and Ac-227. The analysis of U-234 included Th-230 and Ra-226 while the analysis of U-238 included U-234, Th-230, and Ra-226. A basic assumption in using GWSCREEN with radiological progeny is that the progeny travel at the same rate as the parent nuclide with the same retardation or lack of retardation; therefore, simulated peak groundwater concentrations for progeny occur at the same time as the peak parent concentration, which is not always the case in nature. Slope factors for the PBF-22 COC progeny are included in the results tables (Tables 9 and 12).

Another consideration in this risk analysis was the comparison of resultant 100-year receptor groundwater concentrations to the MCLs of EPA's drinking water standards. It is possible, due to differences in current carcinogenic effects data and drinking water protection objectives, to have risk-based concentrations for certain COCs that are in excess of MCLs. Therefore, it is good practice to include MCLs in the risk analysis. Where available, MCLs for the COCs of this analysis are given in Table 6 as well as the risk results tables (Tables 9 through 14). No MCLs were found for Arochlor-1248, Arochlor-1254, Ag, or Zn.

Table 8 lists the assumed receptor reasonable maximum exposure data for the 100-year residential scenario. The modeled receptor is a person consuming 2 L of water per day from a well located at the downgradient edge of the source zone, which is the maximum concentration location. The well's open or screened interval was selected as 15 m (49 ft), according to Track 2 guidance (DOE-ID, 1994b). Receptor intake values such as body weight, averaging time, drinking water intake rate, exposure frequency, and exposure duration were taken from Track 2 guidance (DOE-ID, 1994b). An integration time of 30 years was used for calculating average groundwater concentration.

Table 8. Track 2 default values for other risk input parameters.

Property	GWSCREEN Variable	Default Value	Units
Receptor distance from source parallel to groundwater flow	XD	1/2 of source length	m
Receptor distance from source perpendicular to groundwater flow	YD	0	m
Receptor distance from source depth to aquifer	ZD	0	m
Receptor body weight	BW	70	kg
Averaging time	AT	25,560	d
Receptor water intake rate	WI	2	L/d
Receptor exposure frequency	EF	350	d/yr
Receptor exposure duration	ED	30	yr
Acceptable carcinogenic risk	CRISK	1E-07	
Hazard quotient	HQ	0.1	
Integration time for average	INTIME	30	yr

Since exposure is expressed in terms of intake, the amount of a contaminant taken into the receptor body must be determined. For radiological COCs, this is the total activity ingested. For non-radiological COCs, intake is quantified as amount of COC per unit body weight per unit time (mg/kg-d). The calculation of the total ingested COC includes an ingestion rate (2 L/d), exposure frequency (350 d/yr), and exposure duration (30 years), as well as the COC groundwater concentration. For non-radiological carcinogens and noncarcinogens, the intake calculation includes the receptor body weight (70 kg) and averaging time (365 d/yr x 70 years for carcinogens and 365 d/yr x 30 years for noncarcinogens). The following equations are used to calculate intake:

Carcinogen: Intake = Cwa x IR x EF x ED/(BW x AT)

Noncarcinogen: Intake = Cwa x IR x EF x ED/(BW x AT)

Radionuclide: Intake = Cwa x IR x EF x ED

where Cwa = COC concentration in groundwater, IR = ingestion rate, EF = exposure frequency, ED = exposure duration, BW = body weight, and AT = averaging time. These equations are conservative in that the source is considered to be infinite and the exposure is constant (DOE-ID, 1994b).

INITIAL SCREENING RESULTS

Initially, risks and hazard quotients were calculated for the maximum COC groundwater concentrations at the receptor location as predicted by GWSCREEN. Results of the initial screening calculations for the groundwater pathway based on the peak groundwater concentration are presented in Table 9 for the radionuclide COCs, Table 10 for other carcinogens, and Table 11 for noncarcinogens.

PBF-05

The results presented in Table 9 indicate that Sr-90, with a peak risk of 1E-06, is the only COC at PBF-05 with a calculated peak risk exceeding the 1E-7 risk-screening criterion. Data in Table 4 indicate that Sr-90 was discharged to this injection well over a 6-year period from 1975 to 1980. The results of the GWSCREEN simulations indicate that the peak groundwater concentration at the receptor occurs approximately 16 years after the Sr-90 was released. Therefore, the peak drinking water concentration at the receptor well location could have occurred between 1991 and 1996. Additionally, the peak concentration of 2.3 pCi/L is nearly one-fourth the value of the MCL for Sr-90.

PBF-15

Examination of the peak concentrations of Cr-III, hydrazine, and Zn simulated with GWSCREEN shown in Tables 10 and 11 reveals that none of the PBF-15 COCs exceed either the risk (1E-07) or hazard quotient (0.1) screening criteria at any time. The maximum concentration of Cr-III at the receptor location is 1.6 μg/L at approximately 54 years following its 1971 release, i.e., 2025. The hazard quotient corresponding to this concentration of Cr-III is 8E-06, well below the screening quotient of 0.1. Zn concentration at the receptor location appears to peak 133 years following its release in 1971, i.e., 2104. The peak concentration and corresponding hazard quotient are only 2E-02 μg/L and 3E-07, respectively. Hydrazine has an extremely low retardation factor; that is, it moves in the subsurface almost at the same rate as water. Results indicate that the hydrazine groundwater concentration at the receptor location peaked at 5.1E-02 μg/L approximately 0.5 years following its 1971 release. The integrated health risk associated with this peak is 8E-08. Therefore, no COCs are identified for further evaluation in the 100-year residential scenario as no unacceptable risks are posed by the peak concentrations.

PBF-22

As seen in Tables 12 and 14, all PBF-22 COCs analyzed for health risks or hazard quotients based on peak groundwater concentrations fall below the risk or hazard quotient screening criteria of 1E-07 and 0.1, respectively, with the exception of U-234 and U-238. U-234 produces a peak concentration of 5.3E-01 pCi/L and a health risk of 5E-07; but, due to radioactive progeny, the total risk is 1E-05. However, this peak occurs approximately 5,515 years following the COC's release. The time of occurrence of the combined parent/progeny health risk of 6E-07 for the U-238 chain also occurs approximately 5,515 years after its release. Arochlor-1254 has a peak concentration of 5.2E-02 μg/L and an associated hazard quotient of 0.071. This peak occurs some 15,650 years following the COC's release. In all cases, these time frames of peak concentration are beyond the scope of

Table 9. Peak concentrations and risks for radionuclides.

Site	Radionuclides	Progeny	MCL (pCi/L)	Peak Concentration (pCi/L)	30-yr Average Concentration at Peak (pCi/L)	Intake (pCi)	Slope Factor (risk/pCi)	Risk	Peak Occurrence (years after release)
PBF-05	Co-60		218	2.04E-02	3.32E-03	6.97E+01	1.89E-11	1E-09	43.2
	Cs-137		119	5.78E-02	4.72E-02	9.91E+02	3.16E-11	3E-08	195
	H-3		20,000	9.80E+01	2.24E+01	4.70E+05	7.15E-14	3E-08	7.2
	Sr-90		8	2.25E+00	8.99E-01	1.89E+04	5.59E-11	1E-06	16.4
PBF-22	Am-241		6.45	4.70E-211	4.70E-211	9.87E-207	3.28E-10	3E-216	2.95E+05
		Np-237	7	7.54E-05	7.54E-05	1.58E+00	2.95E-10	5E-10	2.95E+05
		U-233	13.8	5.67E-05	5.67E-05	1.19E+00	4.48E-11	5E-11	2.95E+05
		Th-229	49.3	5.58E-05	5.58E-05	1.17E+00	3.65E-10	4E-10	2.95E+05
	Co-60		218	1.97E-219	1.17E-219	2.46E-215	1.89E-11	5E-226	9.37E+03
	Cs-137		119	1.91E-220	1.82E-220	3.82E-216	3.16E-11	1E-226	4.36E+04
	Pu-239		64.9	2.09E-05	2.09E-05	4.39E-01	3.16E-10	1E-10	8.57E+04
		U-235	30	3.22E-06	3.22E-06	6.76E-02	4.52E-11	3E-12	8.57E+04
		Pa-231	10.2	2.04E-06	2.04E-06	4.28E-02	1.49E-10	6E-12	8.57E+04
		Ac-227	1.27	2.04E-06	2.04E-05	4.28E-02	3.52E-10	2E-11	8.57E+04
	Sr-90		8	1.93E-25	1.84E-25	3.86E-21	5.59E-11	2E-31	2401
	U-234		13.9	5.30E-01	5.27E-01	1.11E+04	4.44E-11	5E-07	5515
		Th-230	79.2	2.97E+00	2.96E+00	6.22E+04	3.75E-11	2E-06	5515
		Ra-226	20	1.86E+00	1.85E+00	3.89E+04	2.95E-10	1E-05	5515
	U-238		14.6	2.27E-01	2.25E-01	4.73E+03	4.27E-11	2E-07	5515
		U-234	13.9	4.04E-01	4.01E-01	8.42E+03	4.44E-11	3E-07	5515
		Th-230	79.2	9.88E-03	9.83E-03	2.06E+02	3.75E-11	8E-09	5515
		Ra-226	20	4.77E-03	4.75E-03	9.98E+01	2.95E-10	3E-08	5515

Table 10. Peak concentrations and risks for other carcinogens.

Site	Other Carcinogens	MCL (μg/L)	Peak Concentration (μg/L)	30-yr Average Concentration at Peak (μg/L)	Intake (mg/kg-d)	Slope Factor (mg/kg-d)$^{-1}$	Risk	Peak Occurrence (years after release)
PBF-15	Hydrazine	400	5.13E-02	2.30E-03	2.70E-08	3.0	8E-08	0.54

Table 11. Peak concentrations and hazard quotients for noncarcinogens.

Site	Noncarcinogens	MCL (μg/L)	Peak Concentration (μg/L)	30-yr Average Concentration at Peak (μg/L)	Intake (mg/kg-d)	Reference Dose (mg/kg/d)	Hazard Quotient	Peak Occurrence (years after release)
PBF-15	Cr-III	100	1.57E+00	2.88E-01	7.89E-06	1.0	7.9E-06	54
	Zn	ND	2.15E-02	3.59E-03	9.84E-08	0.3	3.3E-07	133
PBF-22	Ag	ND	5.96E-02	5.96E-02	1.63E-06	5E-03	3.3E-04	1.45E+04
	Arochlor-1248	ND	1.61E-03	1.60E-03	4.38E-08	ND	not analyzed	1.17E+04
	Arochlor-1254	ND	5.22E-02	5.21E-02	1.43E-06	2E-05	7.1E-02	1.57E+04
	Cr-III	100	1.42E+01	1.33E+01	3.64E-04	1.0	3.6E-04	1.09E+04
	Hg	2	6.78E-03	6.78E-03	1.86E-07	3E-04	6.2E-04	1.61E+04

ND = no toxicity data available.

this analysis. No toxicity data are available for Arochlor-1248; therefore, this COC was not quantitatively assessed in this risk assessment. However, this COC peak groundwater concentration of 1.6E-3 μg/L at the receptor location is less than the peak Arochlor-1254 concentration and occurs about 11,700 years after its release. If the Arochlor-1254 reference dose is assumed for Arochlor-1248, then the resulting hazard quotient will be even less for Arochlor-1248 than the 0.071 determined for Arochlor-1254. Although U-234 and U-238 require further examination, no other COCs identified at PBF-22 are considered for further evaluation in the 100-year residential scenario since no unacceptable risks are posed by their peak groundwater concentrations.

100-YEAR RESIDENTIAL HEALTH RISK ANALYSIS

PBF-05's Sr-90, PBF-22's U-234 (and daughters Th-230 and Ra-226), and PBF-22's U-238 (and daughter U-234) were the only COCs identified from the initial screening for further evaluation in the 100-year residential scenario. This scenario is similar to the initial screening scenario in assuming that the receptor is a 70-kg person who consumes groundwater at a rate of 2 L/day for 350 d/yr for 30 years from a well located at the downgradient edge of the COC source. However, the 100-year residential scenario analyzes risks or hazards based only on a 30-year averaged groundwater concentration that occurs 100 years from the present (i.e., the year 2096), which is assumed to be when institutional control of the site is terminated.

The GWSCREEN input deck for the initial screening assessment of Sr-90 at PBF-05 was modified to provide groundwater concentration and risk output over time and to provide 30-year averaged groundwater concentrations and associated risks. This COC was initially released via the PBF-05 injection well in 1975. To obtain the 100-year residential health risk, the 30-year averaged groundwater concentration was found for the time occurring 121 years after its release (i.e., 2096, 100 years from present). The groundwater concentration at that date was found to be 2.42E-03 pCi/L. This is well below the current Sr-90 MCL of 8 pCi/L. The corresponding health risk is 3E-9. This is below the cumulative risk assessment guidance individual COC screening criterion of 1E-7; therefore, Sr-90 at PBF-05 is not recommended for inclusion in any other further risk analyses. For PBF-22, U-234 and U-238 and their progeny do not appear in the aquifer at any significant concentration at 100 years from present. These PBF-22 COCs are also not recommended for any further risk analyses.

The results of this 100-year residential health risk analysis are presented in Table 12 for radionuclides, Table 13 for other carcinogens, and Table 14 for noncarcinogens. The risks and hazard quotients for other COCs discussed in this report that occur 100 years following their release are also presented. These results indicate that there are no unacceptable groundwater ingestion risks for the 100-year residential scenario. Appendix D contains the GWSCREEN output for Sr-90 (PBF-05), U-234 (PBF-22), and U-238 (PBF-22).

SENSITIVITY/UNCERTAINTY ANALYSIS

Sources of uncertainty in the fate and transport modeling calculations include uncertainty in parameter input values, uncertainty in the description of the subsurface system and release/transport processes, and uncertainty in the waste inventory. In reality, model input parameters are not single

Table 12. 100-year concentrations and risks for radionuclides.

Site	Radionuclides	Progeny	MCL (pCi/L)	30-yr Average Concentration at 100 years (pCi/L)	Intake (pCi)	Slope Factor (risk/pCi)	Risk
PBF-05	Co-60		218	1.78E-08	3.78E-04	1.89E-11	7E-15
	Cs-137		119	0.00	0.00	3.16E-11	0.00
	H-3		20,000	6.63E-17	1.39E-12	7.15E-14	1E-25
	Sr-90		8	2.42E-03	5.08E+01	5.59E-11	3E-09
PBF-22	Am-241		6.45	0.00	9.88E-207	3.28E-10	0.00
		Np-237	7	0.00	1.58E+00	2.95E-10	0.00
		U-233	13.8	0.00	1.19E+00	4.48E-11	0.00
		Th-229	49.3	0.00	1.17E+00	3.65E-10	0.00
	Co-60		218	0.00	2.46E-215	1.89E-11	0.00
	Cs-137		119	0.00	3.82E-216	3.16E-11	0.00
	Pu-239		64.9	0.00	4.39E-01	3.16E-10	0.00
		U-235	30	0.00	6.76E-02	4.52E-11	0.00
		Pa-231	10.2	0.00	4.28E-02	1.49E-10	0.00
		Ac-227	1.27	0.00	4.28E-02	3.52E-10	0.00
	Sr-90		8	0.00	3.86E-21	5.59E-11	0.00
	U-234		13.9	0.00	1.11E+04	4.44E-11	0.00
		Th-230	79.2	0.00	6.22E+04	3.75E-11	0.00
		Ra-226	20	0.00	3.89E+04	2.95E-10	0.00
	U-238		14.6	0.00	4.73EE+03	4.27E-11	0.00
		U-234	13.9	0.00	8.42E+03	4.44E-11	0.00
		Th-230	79.2	0.00	2.06E+02	3.75E-11	0.00
		Ra-226	20	0.00	9.98E+01	2.95E-10	0.00

Table 13. 100-year concentrations and risks for other carcinogens.

Site	Carcinogen	MCL (μg/L)	30-yr Average Concentration at 100 years (μg/L)	Intake (mg/kg-d)	Slope Factor $(mg/kg\text{-}d)^{-1}$	Risk
PBF-15	Hydrazine	400	5.93E-19	6.96E-24	3.0	2E-23

Table 14. 100-year concentrations and hazard quotients for noncarcinogens.

Type of COC	Site	COC	MCL (μg/L)	30-yr Average Concentration at 100 years (μg/L)	Intake (mg/kg-d)	Reference Dose (mg/kg/d)	Hazard Quotient
Non-carcinogenic COCs	PBF-15	Cr-III	100	6.12E-03	1.68E-07	1.0	1.7E-07
		Zn	ND	3.59E-03	9.84E-08	0.3	3.3E-07
	PBF-22	Ag	ND	0	0	5E-03	0
		Arochlor-1248	ND	0	0	ND	0
		Arochlor-1254	ND	0	0	2E-05	0
		Cr-III	100	0	0	1.0	0
		Hg	2	0	0	3E-04	0

values, but vary spatially and temporally over a range of possible values. There is a general lack of site-specific data for use in modeling contaminant fate and transport. Many of the hydrologic and geologic parameters have been inferred from nearby sites (e.g., RWMC), resulting in uncertainty regarding the applicability of the parameters to the PBF area. There is also uncertainty in describing the flow of water and transport contaminants through the complex, fractured rock and soil subsurface. Uncertainty exists also regarding the exact amounts and distribution of contaminants in the PBF source areas.

To account for uncertainties, conservative assumptions and parameters were used where appropriate in an attempt to bound the estimated concentrations. However, an estimate of the uncertainty associated with the predicted concentrations is not feasible since the uncertainty in key model assumptions and model input parameters has not been quantified. Table 15 lists various parameters and assumptions that contribute to the overall uncertainty of the modeling predictions. For each parameter/assumption, a relative degree of conservatism has been assigned from the possibilities: none, low, high, or unknown. A conservatism of "none" implies that the parameter was assigned a value that was reasonable and realistic. A conservatism of "high" implies that the assumption or values assigned the input parameter are expected to produce conservative results.

RECOMMENDATIONS

As a result of the initial peak concentration and the 100-year residential health risk analyses, it is recommended that none of the contaminants associated with any of the three sites, PBF-05, PBF-15, PBF-22, be considered further in any comprehensive risk assessment. There are still some questions regarding appropriate values for sorption coefficients for Arochlor-1254, Arochlor-1248, and hydrazine, as well as the initial contaminant inventories for PBF-22. A sensitivity analysis on the use of the buried source model for the PBF-22 leach pond may prove beneficial. Reasonable discharge rates could be assumed, data input to the pond release model of GWSCREEN, and results compared to those obtained here. It may prove that Arochlor-1254 and -1248 retard so significantly that using the pond release model with conservative pond flux estimates will not cause the resulting groundwater concentrations to be any higher. If the resulting concentrations are higher, then a limiting discharge rate could be back-calculated and compared with known discharge rates for other PBF-area leach ponds.

To bound the uncertainty related to PBF-22 contaminant inventories, GWSCREEN could be used to prepare limiting soil concentrations for each COC. The resulting inventories could then be compared to the by-products of known processes occurring elsewhere at the INEL during the PBF-22 operation to determine if those inventories are reasonable or unrealistic.

Table 15. Uncertainty factors in fate and transport modeling.

Parameter/ assumption	Relative degree of conservatism	Explanation/comment
Pond assumption	Unknown	The pond assumption for the injection wells allows for a more realistic release than the conventional method in GWSCREEN of leaching, but uncertainty regarding the formation around the injection zone and variability in discharge, which leads to uncertainty in the size of the source, makes the conservativeness of the assumption difficult to estimate. Not using the pond assumption for the leach pond is probably not conservative, but it could not be justified without discharge data.
Infiltration rate	Low	The assigned value (10 cm/yr) is the upper end of reasonable range estimated at depth beneath the Subsurface Disposal Area, a similar site.
Discharge rates and inventory	None-Unknown	Discharge rates of water and contaminants for PBF-05 and PBF-22 were obtained from RWMIS and INWMIS data. Sampling results were used to estimate inventories for PBF-22. Waste inventories derived from sampling data are not conclusive, and the reliability is considered low.
Mass homogeneously distributed in source volume (PBF-22 only) at maximum detected concentration	High	Assumption results in maximum contact between infiltrating water and contaminants. Actual contact area will be less.
Source area	None (PBF-22), Unknown (PBF-05 and PBF-15)	The actual pond area was used for PBF-22. The source areas for the injection wells were more conservative than the previous analysis, but probably more representative of the actual size.
Source depth/thickness	None-High	The source thickness for PBF-22 was the average thickness of surficial sediments in the area. The source thickness for the injection wells was chosen as 1 m, which results in equilibrium conditions being established in a relatively short amount of time except for contaminants with high sorption coefficients. This results in the maximum possible release to the unsaturated zone.
Receptor location	High	The receptor is located at the point of maximum expected concentration.
Equilibrium partitioning in source	High	Ignoring kinetic effects associated with transient infiltration results in maximum release.
Solubility limited release	High	Concentrations were assumed not to be limited by solubility constraints.

Table 15. (continued).

Parameter/ assumption	Relative degree of conservatism	Explanation/comment
Plug flow in vadose zone (no dispersion)	High	Longitudinal and transverse spreading (dispersion) of contaminants in vadose zone would result in lower groundwater concentrations.
Vadose zone sediment thickness	None	Assigned thickness was average of total interbed thickness of several wells in the vicinity of the PBF sites.
Neglecting water movement in basalts	High	Additional water travel time and chemical reactions with alteration products in basalt fractures would delay contaminant releases to the aquifer and decrease concentrations. However, differences in travel times may be small compared to the differences caused by uncertainty in the distribution coefficients for the vadose zone sediments.
Moisture content in vadose zone sediments	None	Assigned values are reasonable during operation of the injection wells. The values may be slightly non-conservative after operation ceases because they cause the travel time to be longer than it would under normal conditions. For decaying contaminants, this gives the contaminant more time to decay before reaching groundwater. The difference, however, is small compared to differences caused by uncertainty in the distribution coefficients. For non-decaying contaminants, travel time through the vadose zone does not affect groundwater concentrations in GWSCREEN, only "peak times."
Distribution coefficients	Low-High	Contaminant-specific. In GWSCREEN, the source zone distribution coefficient affects contaminant release rates and thus groundwater concentrations. The vadose zone distribution coefficient affects contaminant travel times through the vadose zone, but not the groundwater concentration because of the plug flow assumption. Concentrations in the aquifer are relatively insensitive to distribution coefficients because the receptor is so close to the source area.
Aquifer porosity	None	The estimated range of values is narrow, and an intermediate value was used.
Groundwater velocity	None	The assigned value is reasonable for the southern portion of the INEL (Wood and Wylie, 1991).
Effective well screen thickness (mixing depth)	None	Mixing depth is reasonable for receptors at the edge of the source areas. Mixing depth is likely to be greater beyond this distance.
Sediment and basalt bulk density	None	Density values are known quite well and have a narrow possible range.

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